4.0 BIOLOGICAL INFORMATION

4.1 LIFE HISTORIES, FACTORS FOR DECLINE, AND CURRENT RANGE-WIDE STATUS

NMFS published the information in this section previously as Appendix A to the paper "A Standardized Quantitative Analysis of the Risks Faced by Salmonids in the Columbia River Basin" (McClure et al. 2000). Additional details regarding the life histories, factors for decline, and current range-wide status of these species are available in Appendix A of this Biological Opinion.

4.1.1 Snake River Spring/Summer Chinook Salmon

4.1.1.1 Geographic Boundaries and Spatial Distribution

The location, geology, and climate of the Snake River region create a unique aquatic ecosystem for chinook salmon. Spring- and/or summer-run chinook salmon are found in five principal subbasins of the Snake River (CBFWA 1990). Of these, the Grande Ronde and Salmon rivers are large, complex systems composed of several smaller tributaries that are further composed of many small streams. In contrast, the Tucannon and Imnaha rivers are small systems with most salmon production in the main river. In addition to the five major subbasins, three small streams (Asotin, Granite, and Sheep creeks) that enter the Snake River between Lower Granite and Hells Canyon dams provide small spawning and rearing areas (CBFWA 1990). Although there are some indications that more than one ESU may exist within the Snake River basin, the available data are not sufficient to clearly demonstrate the existence of multiple ESUs or to define their boundaries. Because of compelling genetic and life-history evidence that fall chinook salmon are distinct from other chinook salmon in the Snake River, however, they are considered a separate ESU.

4.1.1.2 Historical Information

Historically, spring and/or summer chinook salmon spawned in virtually all accessible and suitable habitat in the Snake River system (Evermann 1896; Fulton 1968). During the late 1800s, the Snake River produced a substantial fraction of all Columbia River basin spring and summer chinook salmon, with total production probably exceeding 1.5 million in some years. By the mid-1900s, the abundance of adult spring and summer chinook salmon had greatly declined. Fulton (1968) estimated that an average of 125,000 adults per year entered the Snake River tributaries from 1950 through 1960. As evidenced by adult counts at dams, however, spring and summer chinook salmon have declined considerably since the 1960s (Corps 1989).

4.1.1.3 Life History

In the Snake River, spring and summer chinook share key life history traits. Both are stream-type fish, with juveniles that migrate swiftly to sea as yearling smolts. Depending primarily on location within the basin (and not on run type), adults tend to return after either 2 or 3 years in the ocean. Both spawn and rear in small, high-elevation streams (Chapman et al. 1991), although where the two forms co-exist, spring-run chinook spawn earlier and at higher elevations than summer-run chinook.

4.1.1.4 Habitat and Hydrology

Even before mainstem dams were built, habitat was lost or severely damaged in small tributaries by construction and operation of irrigation dams and diversions, inundation of spawning areas by impoundments, and siltation and pollution from sewage, farming, logging, and mining (Fulton 1968). Recently, the construction of hydroelectric and water storage dams without adequate provision for adult and juvenile passage in the upper Snake River has kept fish from all spawning areas upstream of Hells Canyon Dam.

4.1.1.5 Hatchery Influence

There is a long history of human efforts to enhance production of chinook salmon in the Snake River basin through supplementation and stock transfers. The evidence is mixed as to whether these efforts have altered the genetic makeup of indigenous populations. Straying rates appear to be very low.

4.1.1.6 Population Trends and Risks

For the SR spring/summer chinook salmon ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period[1] ranges from 0.94 to 0.66, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to the effectiveness of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated average population growth rates and the risk of absolute extinction within 24 and 100 years for six of the seven spring/summer chinook salmon index stocks, using the same range of assumptions about the relative effectiveness of hatchery fish (no data were available on the proportion of hatchery fish in the seventh population, Bear Creek). At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years for the wild component ranges from 0.01 for Johnson Creek to 1.00 for the Imnaha River (Table A-6a; Appendix B in McClure et al. 2000). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is

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[1] The estimates of average population growth rate, risk of extinction, and likelihood of meeting recovery goals are based on population trends observed during the base period of 1980 through 1994 (including 1999 adult returns).

1.00 for the wild components in the Imnaha and Minam rivers (Table A-6d; Appendix B in McClure et al. 2000).

NMFS has also calculated the proportional increase in the average growth rates of the SR spring/summer chinook salmon index stocks that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-7a through A-7d; Appendix B in McClure et al. 2000). This analysis explored the sensitivity of the results to different methods of projecting population trends 100 years into the future. One of the projections was based on the observed returns from the 1980 through 1994 brood years ("Observed") whereas two also projected returns for the 1995 through 1996 ("Projected-1") and 1995 through 1999 ("Projected-2") brood years, respectively. Assuming that the effectiveness of hatchery fish has been zero, a relatively small change (less than 15%) is needed in the growth rate of the wild population for each of the index populations under all three projections. However, as the relative effectiveness of hatchery-origin spawners increases, the needed change in growth rate also rises, to more than 100% for the Minam River if hatchery-origin spawners have been 100% as effective as wild fish (Table A-7d).

Similar results are shown in Tables A-8a and A-8b for the population growth rate needed for recovery of SR spring/summer chinook salmon index populations. The sensitivity of the recovery metrics to different assumptions methods of projecting population trends into the future is shown in Tables A9a to A9b.

4.1.2 Snake River Fall Chinook Salmon

4.1.2.1 Geographic Boundaries and Spatial Distribution

The Snake River basin drains an area of approximately 280,000 km^2, and incorporates a range of vegetative life zones, climatic regions, and geological formations, including the deepest canyon (Hells Canyon) in North America. The ESU includes the mainstem river and all tributaries, from their confluence with the Columbia River to the Hells Canyon complex. Because genetic analyses indicate that fall-run chinook salmon in the Snake River are distinct from the spring/summer-run in the Snake basin (Waples et al. 1991), SR fall chinook salmon are considered separately from the other two forms. They are also considered separately from those assigned to the UCR summer- and fall-run ESU because of considerable differences in habitat characteristics and adult ocean distribution and less definitive, but still significant, genetic differences. There is, however, some concern that recent introgression from Columbia River hatchery strays is causing the Snake River population to lose the qualities that made it distinct for ESA purposes.

4.1.2.2 Historical Information

SR fall chinook salmon remained stable at high levels of abundance through the first part of the twentieth century, but then declined substantially. Although historical abundance of fall chinook salmon in the Snake River is difficult to estimate, adult returns appear to have declined by three orders of magnitude since the 1940s, and perhaps by another order of magnitude from pristine levels. Irving and Bjornn (1981b) estimated that the mean number of fall chinook salmon returning to the Snake River declined from 72,000 from 1938 to 1949 to 29,000 during the 1950s. Further declines occurred upon completion of the Hells Canyon complex, which blocked access to primary production areas in the late 1950s (see below).

4.1.2.3 Life History

Fall chinook salmon in this ESU are ocean-type. Adults return to the Snake River at ages 2 through 5, with age 4 most common at spawning (Chapman et al. 1991). Spawning, which takes place in late fall, occurs in the mainstem and in the lower parts of major tributaries (NWPPC 1989; Bugert et al. 1990). Juvenile fall chinook salmon move seaward slowly as subyearlings, typically within several weeks of emergence (Chapman et al. 1991). Coded wire-tag data indicate that SR fall chinook have a far-ranging ocean distribution, with a significant proportion (about 20%) taken in Alaska and Canada, and in recent years 97% off Washington, Oregon, and California (Pacific Salmon Commission, Chinook Technical Committee Model 2000).

4.1.2.4 Habitat and Hydrology

With hydrosystem development, the most productive areas of the Snake River basin are now inaccessible or inundated. The upper reaches of the mainstem Snake River were the primary areas used by fall chinook salmon, with only limited spawning activity reported downstream from Rkm 439. The construction of Brownlee Dam (1958; Rkm 459), Oxbow Dam (1961; Rkm 439), and Hells Canyon Dam (1967; Rkm 397 eliminated the primary production areas of SR fall chinook salmon. There are now 12 dams on the mainstem Snake River, and they have substantially reduced the distribution and abundance of fall chinook salmon (Irving and Bjornn 1981a).

4.1.2.5 Hatchery Influence

The Snake River has contained hatchery-reared fall chinook salmon since 1981 (Busack 1991b). The hatchery contribution to Snake River escapement has been estimated at greater than 47 percent (NMFS 1998). Artificial propagation is recent, so cumulative genetic changes associated with it may be limited. Wild fish are incorporated into the brood stock each year, which should reduce divergence from the wild population. Release of subyearling fish may also help minimize the differences in mortality patterns between hatchery and wild populations that can lead to genetic change (Waples in press).

4.1.2.6 Other

Some SR fall chinook historically migrated over 1,500 km from the ocean. Although the Snake River population is now restricted to habitat in the lower river, genes associated with the lengthier migration may still reside in the population. Because longer freshwater migrations in chinook salmon tend to be associated with more-extensive oceanic migrations (Healey 1983), maintaining populations occupying habitat that is well inland may be important in maintaining diversity in the marine ecosystem as well.

4.1.2.7 Population Trends and Risks

For the SR fall chinook salmon ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.93 to 0.59, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction within 24 and 100 years for the aggregate SR fall chinook population, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years is 0.50 (Table A-6a). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 (Table A-6d).

NMFS has also calculated the proportional increase in the average growth rate of the aggregate SR fall chinook salmon population that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, a relatively small change (4%) is needed in the growth rate of the wild population (Table A-6a). The needed change in growth rate rises to 65% if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

Similar results are shown in Tables A-8a and A-8b for the population growth rate needed for recovery of the aggregate SR fall chinook salmon population. The sensitivity of the recovery metrics to different assumptions methods of projecting population trends into the future is shown in Tables A9a to A9b.

4.1.3 Upper Columbia River Spring-run Chinook Salmon

4.1.3.1 Geographic Boundaries and Spatial Distribution

This ESU includes spring-run chinook populations found in Columbia River tributaries between the Rock Island and Chief Joseph dams, notably the Wenatchee, Entiat, and Methow River basins. The populations are genetically and ecologically separate from the summer- and fall-run populations in the lower parts of many of the same river systems (Myers et al. 1998). Although fish in this ESU are genetically similar to spring chinook in adjacent ESUs (i.e., Mid-Columbia and Snake), they are distinguished by ecological differences in spawning and rearing habitat preferences. For example, spring-run chinook in upper Columbia tributaries spawn at lower elevations (500 to 1,000 m) than in the Snake and John Day River systems.

4.1.3.2 Historical Information

The upper Columbia River populations were intermixed during the Grand Coulee Fish Maintenance Project (1939 through 1943), resulting in loss of genetic diversity between populations in the ESU. Homogenization remains an important feature of the ESU. Fish abundance has trended downward both recently and over the long term. At least six former populations from this ESU are now extinct, and nearly all extant populations have fewer than 100 wild spawners.

4.1.3.3 Life History (Including Ocean)

UCR spring chinook are considered stream-type fish, with smolts migrating as yearlings. Most stream-type fish mature at 4 years of age. Few coded-wire tags are recovered in ocean fisheries, suggesting that the fish move quickly out of the north central Pacific and do not migrate along the coast.

4.1.3.4 Habitat and Hydrology

Salmon in this ESU must pass up to nine Federal and private dams, and Chief Joseph Dam prevents access to historical spawning grounds farther upstream. Degradation of remaining spawning and rearing habitat continues to be a major concern associated with urbanization, irrigation projects, and livestock grazing along riparian corridors. Overall harvest rates are low for this ESU, currently less than 10% (ODFW and WDFW 1995).

4.1.3.5 Hatchery Influence

Spring-run chinook salmon from the Carson National Fish Hatchery (a large composite, non-native stock) were introduced into and have been released from local hatcheries (Leavenworth, Entiat, and Winthrop National Fish Hatcheries [NFH]). Little evidence suggests that these hatchery fish stray into wild areas or hybridize with naturally spawning populations. In addition to these national production hatcheries, two supplementation hatcheries are operated by the Washington Department of Fish and Wildlife (WDFW) in this ESU. The Methow Fish Hatchery Complex (MFC 1992) and the Rock Island Fish Hatchery Complex (RIFH 1989) were both designed to implement supplementation programs for naturally spawning populations on the Methow and Wenatchee rivers, respectively (Chapman et al. 1995).

4.1.3.6 Population Trends and Risks

For the UCR spring-run chinook salmon ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.87 to 0.78, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated average population growth rates and the risk of absolute extinction within 24 and 100 years for the three spawning populations identified by Ford et al. (1999), using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years ranges from 0.73 for the Methow River to 1.00 for the Methow and Entiat rivers (Table A-6a; Appendix B in McClure et al. 2000). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of extinction within 100 years is 1.00 for all three spawning populations (Table A-6d; Appendix B in McClure et al. 2000).

NMFS has also calculated the proportional increase in the average growth rates of each population that would be needed to reduce the risk of absolute extinction within 100 years to five% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, relatively small changes ($\leq$20%) are needed in the growth rates of each of the wild populations (Table A-6a). The needed change in growth rate rises as high as 60% for the Entiat River population if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

Similar results are shown in Tables A-8a and A-8b for the population growth rate needed for recovery of the Wenatchee River spring chinook salmon population. The sensitivity of the recovery metrics to different assumptions methods of projecting population trends into the future is shown in Tables A9a to A9b.

NMFS has also used population risk assessments for Upper Columbia spring chinook and steelhead ESU's from the draft Quantitative Analysis Report (Cooney, 2000 DRAFT). Risk assessments described in that report were based on Monte Carlo simulations with simple spawner/spawner models that incorporate estimated smolt carrying capacity. Population dynamics were simulated for three separate spawning populations in the UCR spring chinook salmon ESU, the Wenatchee, Entiat and Methow populations. The QAR assessments showed extinction risks for UCR spring chinook salmon of 61% for the Methow and 99% for the Entiat spawning populations (Table A6e). These estimates are based on the assumption that the median return rate for the 1980 brood year to the 1994 brood year series will continue into the future.

The QAR analyses also include estimates of the percent change in the spawner replacement rate necessary to meet survival and recovery criteria. For UCR spring chinook salmon, the percent change in survival necessary to reduce the risk of absolute extinction to less than 5% within 100 years ranged from 29% for the Methow population to 75% for the Wenatchee (Table A-10). Substantial improvements in survival are needed to meet recovery levels for all three spawning populations. The estimated survival changes required to meet or exceed the interim Recovery Criteria established by the Upper Columbia River Sockeye, Steelhead, and Chinook Salmon Biological Requirements Committee ranged from 95% (Methow) to 175% (Wenatchee).

4.1.4 Upper Willamette River Chinook Salmon

4.1.4.1 Geographic Boundaries and Spatial Distribution

The UWR chinook ESU includes native spring-run populations above Willamette Falls and in the Clackamas River. In the past, it included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River, and Albiqua Creek. Although the total number of fish returning to the Willamette has been relatively high (24,000), about 4,000 fish now spawn naturally in the ESU, two-thirds of which originate in hatcheries. The McKenzie River supports the only remaining naturally reproducing population in the ESU.

4.1.4.2 Historical Information

There are no direct estimates of the size of the chinook salmon runs in the Willamette River basin before the 1940s. McKernan and Mattson (1950) present anecdotal information that the native American fishery at the Willamette Falls may have yielded 2,000,000 lb (908,000 kg) of salmon (454,000 fish, each weighing 20 lb [9.08 kg]). Based on egg collections at salmon hatcheries, Mattson (1948) estimates that the spring chinook salmon run in the 1920s may have been 5 times the run size of 55,000 fish (in 1947), or 275,000 fish. Much of the early information on salmon runs in the upper Willamette River basin comes from operation reports of state and Federal hatcheries.

4.1.4.3 Life History

Fish in this ESU are distinct from those of adjacent ESUs in life history and marine distribution. UWR chinook salmon have an ocean-type life history, migrating to the ocean during their first autumn. Coded-wire-tag recoveries indicate that the fish travel to the marine waters of British Columbia and Alaska. More Willamette fish are, however, recovered in Alaskan waters than fish from the Lower Columbia River ESU. UWR chinook mature in their fourth or fifth years. Historically, 5-year-old fish dominated the spawning migration runs, but recently, most fish have matured at age 4. The timing of the spawning migration is limited by Willamette Falls. High flows in the spring allow access to the upper Willamette basin, whereas low flows in the summer and autumn prevent later-migrating fish from ascending the falls. The low flows may serve as an isolating mechanism, separating this ESU from others nearby.

4.1.4.4 Habitat and Hydrology

Human activities have had enormous effects on the salmonid populations in the Willamette drainage. First, the Willamette River, once a highly braided river system, has been dramatically simplified through channelization, dredging, and other activities that have reduced rearing habitat (i.e., stream shoreline) by as much as 75%. In addition, the construction of 37 dams in the basin has blocked access to over 700 km of stream and river spawning habitat. The dams also alter the temperature regime of the Willamette and its tributaries, affecting the timing of development of naturally spawned eggs and fry. Water quality is also affected by development and other economic activities. Agricultural and urban land uses on the valley floor, as well as timber harvesting in the Cascade and Coast ranges, contribute to increased erosion and sediment load in Willamette basin streams and rivers. Finally, since at least the 1920s, the lower Willamette has suffered municipal and industrial pollution.

4.1.4.5 Hatchery Influence

Hatchery production in the basin began in the late nineteenth century. Eggs were transported throughout the basin, resulting in current populations that are relatively homogeneous genetically (although still distinct from those of surrounding ESUs). Hatchery production continues in the Willamette, with an average of 8.4 million smolts and fingerlings released each year into the main river or its tributaries between 1975 and 1994. Hatcheries are currently responsible for most production (90% of escapement) in the basin.

4.1.4.6 Other

Harvest on this ESU is high, both in the ocean and inriver. The total inriver harvest below the falls from 1991 through 1995 averaged 33% and was much higher before then. Ocean harvest was estimated as 16% for 1982 through 1989. ODFW (1998) indicates that total (marine and freshwater) harvest rates on UWR spring-run stocks are reduced considerably for the 1991 through 1993 brood years, to an average of 21%.

4.1.4.7 Population Trends and Risks

For the UWR chinook salmon ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 1.01 to 0.17, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction within 24 and 100 years for the aggregate Upper Columbia River chinook salmon population in the McKenzie River, above Leaburg, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years is 0.02 (Table A-6a). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 (Table A-6d).

NMFS has also calculated the proportional increase in the average growth rate of the aggregate McKenzie River population that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, no change is needed in the growth rate of the wild population (Table A-6a). The needed change in growth rate rises to 87% if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

4.1.5 Lower Columbia River Chinook Salmon

4.1.5.1 Geographic Boundaries and Spatial Distribution

The Lower Columbia River ESU is characterized by numerous short and medium-length rivers that drain the coast ranges and the west slope of the Cascade Mountains. This ESU includes all native populations from the mouth of the Columbia River to the crest of the Cascade Range, excluding populations above Willamette Falls. The former location of Celilo Falls (drowned by The Dalles Reservoir in 1960) is the eastern boundary for this ESU. Stream-type spring-run chinook salmon found in the Klickitat River or the introduced Carson spring-chinook salmon strain are not included in this ESU. Spring-run chinook salmon in the Sandy River have been influenced by spring-run chinook salmon introduced from the Willamette River ESU. However, analyses suggest that considerable genetic resources still reside in the existing population (NMFS 1998).

4.1.5.2 Historical Information

Historical records of chinook salmon abundance are sparse, but cannery records suggest a peak run of 4.6 million fish in 1883. Although fall-run chinook salmon are still present throughout much of their historical range, most of the fish spawning today are first-generation hatchery strays. Furthermore, spring-run populations have been severely depleted throughout the ESU and extirpated from several rivers.

4.1.5.3 Life History

Fish in this ESU are considered ocean-type; most emigrate to the ocean as subyearlings. Coded wire-tag recoveries of Lower Columbia River ESU fish suggest a northerly migration route, but the fish apparently contribute little to the Alaskan fishery. Populations tend to mature at ages 3 and 4.

4.1.5.4 Habitat and Hydrology

As in other ESUs, chinook salmon have been affected by the alteration of freshwater habitat (Bottom et al. 1985, WDF et al. 1993, Kostow 1995). Timber harvesting and associated road building peaked in the 1930s, but effects from the timber industry remain (Kostow 1995). Agriculture is widespread in this ESU and has affected riparian vegetation and stream hydrology. The ESU is also highly affected by urbanization, including river diking and channelization, wetland draining and filling, and pollution (Kostow 1995).

4.1.5.5 Hatchery Influence

The Lower Columbia River ESU has been subject to intensive hatchery influence. Hatchery programs to enhance chinook salmon fisheries in the lower Columbia River began in the 1870s, releasing billions of fish over time. That equals the total hatchery releases for all other chinook ESUs combined (Myers et al. 1998). Although most of the stocks have come from inside the ESU, more than 200 million fish from outside the ESU have been released since 1930 (Myers et al. 1998).

4.1.5.6 Population Trends and Risks

For the LCR chinook salmon ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.95 to 0.62, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction within 24 and 100 years for ten subbasin populations, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years ranges from 0.03 for Plympton Creek to 1.00 for Bear and Mill creeks and the Klaskanine and Youngs rivers (Table A-6a). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 for all but one of the 10 subbasin populations (0.82 for the Sandy River late run; Table A-6d). The Lewis and Clark River population is functionally extinct.

NMFS has also calculated the proportional increase in the average growth rate of each subbasin population that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, the needed change in the growth rate of the wild population ranges from zero in Plympton Creek to 112% in the Youngs River (Table A-6a). The needed change in growth rate rises as high as 287% in the Youngs River if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

4.1.6 Snake River Steelhead

4.1.6.1 Geographic Boundaries and Spatial Distribution

Steelhead spawning habitat in the Snake River is distinctive in having large areas of open, low-relief streams at high elevations. In many Snake River tributaries, spawning occurs at a higher elevation (up to 2,000 m) than for steelhead in any other geographic region. SR steelhead also migrate farther from the ocean (up to 1,500 km) than most.

4.1.6.2 Historical Information

No estimates of historical (pre-1960s) abundance specific to this ESU are available.

4.1.6.3 Life History (Including Ocean)

Fish in this ESU are summer steelhead. They enter freshwater from June to October and spawn during the following March to May. Two groups are identified, based on migration timing, ocean-age, and adult size. A-run steelhead, thought to be predominately age-1-ocean, enter freshwater during June through August. B-run steelhead, thought to be age-2-ocean, enter freshwater during August through October. B-run steelhead typically are 75 to 100 mm longer at the same age. Both groups usually smolt as 2- or 3-year-olds (Whitt 1954, BPA 1992, Hassemer 1992).

4.1.6.4 Habitat and Hydrology

Hydrosystem projects create substantial habitat blockages in this ESU; the major ones are the Hells Canyon Dam complex (mainstem Snake River) and Dworshak Dam (North Fork Clearwater River). Minor blockages are common throughout the region. Steelhead spawning areas have been degraded by overgrazing, as well as by historical gold dredging and sedimentation due to poor land management. Habitat in the Snake basin is warmer and drier and often more eroded than elsewhere in the Columbia River basin or in coastal areas.

4.1.6.5 Hatchery Influence

Hatchery fish are widespread and stray to spawn naturally throughout the region. In the 1990s, an average of 86% of adult steelhead passing Lower Granite Dam were of hatchery origin. Hatchery contribution to naturally spawning populations varies, however, across the region. Hatchery fish dominate some stocks, whereas wild fish compose all of others.

4.1.6.6 Population Trends and Risks

For the SR steelhead ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.90 to 0.18, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction within 24 and 100 years for the A and B runs, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years ranges from 0.12 for A-run steelhead to 0.35 for B-run fish (Table A6-a). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 for both runs (Table A-6d).

NMFS has also calculated the proportional increase in the average growth rate of each run that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, the needed change in the growth rate of the wild population ranges from 0.01 for A-run steelhead to 0.02 for the B run (Table A-6a). The maximum needed change in growth rate rises as high as 470% for B-run steelhead if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

4.1.7 Upper Columbia River Steelhead

4.1.7.1 Geographic Boundaries and Spatial Distribution

This ESU occupies the Columbia River basin upstream of the Yakima River. Rivers in the area primarily drain the east slope of the northern Cascade Mountains and include the Wenatchee, Entiat, Methow, and Okanogan River basins. The climate of the area reaches temperature and precipitation extremes; most precipitation falls as mountain snow (Mullan et al.1992). The river valleys are deeply dissected and maintain low gradients, except for the extreme headwaters (Franklin and Dyrness 1973).

4.1.7.2 Historical Information

Estimates of historical (pre-1960s) abundance specific to this ESU are available from fish counts at dams. Counts at Rock Island Dam from 1933 to 1959 averaged 2,600 to 3,700, suggesting a pre-fishery run size exceeding 5,000 adults for tributaries above Rock Island Dam (Chapman et al. 1994). Runs may, however, already have been depressed by lower Columbia River fisheries.

4.1.7.3 Life History

As in other inland ESUs (the Snake and mid-Columbia River basins), steelhead in the Upper Columbia River ESU remain in freshwater up to a year before spawning. Smolt age is dominated by 2-year-olds. Based on limited data, steelhead from the Wenatchee and Entiat rivers return to freshwater after 1 year in salt water, whereas Methow River steelhead are primarily age-2-ocean (Howell et al. 1985). Life history characteristics for UCR steelhead are similar to those of other inland steelhead ESUs; however, some of the oldest smolt ages for steelhead, up to 7 years, are reported from this ESU. The relationship between anadromous and nonanadromous forms in the geographic area is unclear.

4.1.7.4 Habitat and Hydrology

The Chief Joseph and Grand Coulee dam construction caused blockages of substantial habitat, as did that of smaller dams on tributary rivers. Habitat issues for this ESU relate mostly to irrigation diversions and hydroelectric dams, as well as to degraded riparian and instream habitat from urbanization and livestock grazing.

4.1.7.5 Hatchery Influence

Hatchery fish are widespread and escape to spawn naturally throughout the region. Spawning escapement is dominated by hatchery-produced fish.

4.1.7.6 Population Trends and Risks

For the UCR steelhead ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.90 to 0.29, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction for the aggregate UCR steelhead population, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years is 0.95 (Table A-6a). Assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 (Table A-6d).

NMFS has also calculated the proportional increase in the average growth rate of the aggregate population that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, a 7% increase would be needed in the growth rate of the wild population (Table A-6a). The needed change in growth rate rises to 225% if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

Similar results are shown in Tables A-8a and A-8b for the population growth rate needed for recovery of the aggregate UCR steelhead population. The sensitivity of the recovery metrics to different assumptions methods of projecting population trends into the future is shown in Tables A9a to A9b.

Due to data limitations, the QAR steelhead assessments were limited to two aggregate spawning groups – the Wenatchee/Entiat composite and the above-Wells populations. Wild production of steelhead above Wells Dam was assumed to be limited to the Methow system. Assuming a relative effectiveness of hatchery spawners of 1.0, the risk of absolute extinction within 100 years for UCR steelhead is 100% (Table A-6f). The QAR also assumed hatchery effectiveness values of 0.25 and 0.75. A hatchery effectiveness of 0.25 resulted in projected risks of extinction of 22% for the Wenatchee/Entiat and 28% for the Methow populations. At a hatchery effectiveness of 0.75, risks of 100% were projected for both populations.

The change in survival necessary to reduce extinction risks to less than 5% within 100 years ranged from 15% for the Methow population, assuming hatchery effectiveness 0.25, to 67% for the Wenatchee/Entiat population (Table A-10). Substantial improvements in survival are needed to meet recovery levels for both spawning populations. The estimated survival changes required to meet or exceed the interim Recovery Criteria established by the Upper Columbia River Sockeye, Steelhead, and Chinook Salmon Biological Requirements Committee ranged from 50% for the Wenatchee/Entiat population at a hatchery spawner effectiveness of 0.25 to 200% for the Methow population at an effectiveness of 0.75.

4.1.8 Middle Columbia River Steelhead

4.1.8.1 Geographic Boundaries and Spatial Distribution

The Middle Columbia River ESU occupies the Columbia River basin from above the Wind River in Washington and the Hood River in Oregon and continues upstream to include the Yakima River, Washington. The region includes some of the driest areas of the Pacific Northwest, generally receiving less than 40 cm of precipitation annually (Jackson 1993). Summer steelhead are widespread throughout the ESU; winter steelhead occur in Mosier, Chenowith, Mill, and Fifteenmile creeks, Oregon, and in the Klickitat and White Salmon rivers, Washington. The John Day River probably represents the largest native, natural spawning stock of steelhead in the region.

4.1.8.2 Historical Information

Estimates of historical (pre-1960s) abundance specific to this ESU are available for the Yakima River, which has an estimated run size of 100,000 (WDF et al. 1993). Assuming comparable run sizes for other drainage areas in this ESU, the total historical run size may have exceeded 300,000 steelhead.

4.1.8.3 Life History

Most fish in this ESU smolt at 2 years and spend 1 to 2 years in salt water before reentering freshwater, where they may remain up to a year before spawning (Howell et al. 1985, BPA 1992). All steelhead upstream of The Dalles Dam are summer-run (Schreck et al. 1986, Reisenbichler et al. 1992, Chapman et al. 1994). The Klickitat River, however, produces both summer and winter steelhead, and age-2-ocean steelhead dominate the summer steelhead, whereas most other rivers in the region produce about equal numbers of both age-1- and 2-ocean fish. A nonanadromous form co-occurs with the anadromous form in this ESU; information suggests that the two forms may not be isolated reproductively, except where barriers are involved.

4.1.8.4 Habitat and Hydrology

The only substantial habitat blockage now present in this ESU is at Pelton Dam on the Deschutes River, but minor blockages occur throughout the region. Water withdrawals and overgrazing have seriously reduced summer flows in the principal summer steelhead spawning and rearing tributaries of the Deschutes River. This is significant because high summer and low winter temperatures are limiting factors for salmonids in many streams in this region (Bottom et al. 1985).

4.1.8.5 Hatchery Influence

Continued increases in the proportion of stray steelhead in the Deschutes River basin is a major concern. The ODFW and CTWSRO estimate that 60 to 80% of the naturally spawning population consists of strays, which greatly outnumber naturally produced fish. Although the reproductive success of stray fish has not been evaluated, their numbers are so high that major genetic and ecological effects on natural populations are possible (Busby et al. 1999).

The negative effects of any interbreeding between stray and native steelhead will be exacerbated if the stray steelhead originated in geographically distant river basins, especially if the river basins are in different ESUs. The populations of steelhead in the Deschutes River basin include the following:

1. Steelhead native to the Deschutes River
2. Hatchery steelhead from the Round Butte Hatchery on the Deschutes River
3. Wild steelhead strays from other rivers in the Columbia River basin
4. Hatchery steelhead strays from other Columbia River basin streams

Regarding the latter, CTWSRO reports preliminary findings from a tagging study by T. Bjornn and M. Jepson (University of Idaho) and NMFS suggesting that a large fraction of the steelhead passing through Columbia River dams (e.g., John Day and Lower Granite dams) have dipped into the Deschutes River and then returned to the mainstem Columbia River. A key unresolved question about the large number of strays in the Deschutes basin is how many stray fish remain in the basin and spawn naturally.

4.1.8.6 Population Trends and Risks

For the MCR steelhead ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.87 to 0.33, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction for four subbasin populations, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years ranges from zero for the Deschutes and Yakima River summer runs to 1.00 for the Umatilla River summer run (Table A-6a). Assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years rises to 1.00 for the Deschutes River summer run as well (Table A-6d).

NMFS has also calculated the proportional increase in the average growth rate of each of the four subbasin populations that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, changes in population growth rates of less than 10 % would be needed in the growth rate of the wild population (Table A-6a). The needed change in growth rate rises to a maximum of 252% for the Deschutes River summer run if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

4.1.9 Upper Willamette River Steelhead

4.1.9.1 Geographic Boundaries and Spatial Distribution

The UWR steelhead ESU occupies the Willamette River and tributaries upstream of Willamette Falls, extending to and including the Calapooia River. These major river basins containing spawning and rearing habitat comprise more than 12,000 km^2 in Oregon. Rivers that contain naturally spawning winter-run steelhead include the Tualatin, Mollala, Santiam, Calapooia, Yamhill, Rickreall, Luckiamute, and Mary's, although the origin and distribution of steelhead in a number of these basins is being debated. Early migrating winter and summer steelhead have been introduced into the upper Willamette River basin, but those components are not part of the ESU.

4.1.9.2 Historical Information

Native winter steelhead within this ESU have been declining since 1971 and have exhibited large fluctuations in abundance.

4.1.9.3 Life History

In general, native steelhead of the upper Willamette River basin are late-migrating winter steelhead, entering freshwater primarily in March and April. This atypical run timing appears to be an adaptation for ascending Willamette Falls, which functions as an isolating mechanism for UWR steelhead. Reproductive isolation resulting from the falls may explain the genetic distinction between steelhead from the upper Willamette River basin and those in the lower river. UWR late-migrating steelhead are ocean-maturing fish. Most return at age 4, with a small proportion returning as 5-year-olds (Busby et al. 1996).

4.1.9.4 Habitat and Hydrology

Willamette Falls (Rkm 77) is a known migration barrier. Winter steelhead and spring chinook salmon historically occurred above the falls, whereas summer steelhead, fall chinook, and coho salmon did not. Detroit and Big Cliff dams cut off 540 km of spawning and rearing habitat in the North Santiam River. In general, habitat in this ESU has become substantially simplified since the 1800s by removal of large woody debris to increase the river's navigability.

4.1.9.5 Hatchery Influence

The main hatchery production of native (late-run) winter steelhead occurs in the North Fork Santiam River, where estimates of hatchery proportion in natural spawning areas range from 14% to 54% (Busby et al. 1996). More recent estimates of the percentage of naturally spawning fish attributable to hatcheries in the late 1990s are 24% in the Molalla, 17% in the North Santiam, 5% to 12% in the South Santiam, and less than 5% in the Calapooia (Chilcote 1997, 1998).

4.1.9.6 Population Trends and Risks

For the UWR steelhead ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.91 to 0.66, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction for four subbasin populations, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years ranges from zero for the South Santiam River to 1.00 for the Calipooia River (Table A-6a). Assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years rises to 1.00 for all four populations (Table A-6d).

NMFS has also calculated the proportional increase in the average growth rate of each of the four subbasin populations that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, changes of up to 26% (for the Calipooia River) would be needed in the growth rates of the wild populations (Table A-6a). The needed change in growth rate rises to a maximum of 88% for the Mollala River if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

4.1.10 Lower Columbia River Steelhead

4.1.10.1 Geographic Boundaries and Spatial Distribution

The Lower Columbia River ESU encompasses all steelhead runs in tributaries between the Cowlitz and Wind rivers on the Washington side of the Columbia, and the Willamette and Hood rivers on the Oregon side. The populations of steelhead that make up the Lower Columbia River ESU are distinguished from adjacent populations by genetic and habitat characteristics. The ESU consists of summer and winter coastal steelhead runs in the tributaries of the Columbia River as it cuts through the Cascades. These populations are genetically distinct from inland populations (east of the Cascades), as well as from steelhead populations in the upper Willamette basin and coastal runs north and south of the Columbia River mouth. Not included in the ESU are runs in the Willamette River above the Clackamas (Upper Willamette River ESU), runs in the Little and Big White Salmon rivers (Middle Columbia River ESU) and runs based on four imported hatchery stocks: early-spawning winter Chambers Creek/lower Columbia River mix, summer Skamania Hatchery stock, winter Eagle Creek NFH stock, and winter Clackamas River ODFW stock (NMFS 1998). This area has at least 36 distinct runs (Busby et al. 1996), 20 of which were identified in the initial listing petition. In addition, numerous small tributaries have historical reports of fish, but no current abundance data. The major runs in the ESU, for which there are estimates of run size, are the Cowlitz River winter runs, Toutle River winter runs, Kalama River winter and summer runs, Lewis River winter and summer runs, Washougal River winter and summer runs, Wind River summer runs, Clackamas River winter and summer runs, Sandy River winter and summer runs, and Hood River winter and summer runs.

4.1.10.2 Historical Information

For the larger runs, current counts have been in the range of one to 2,000 fish (Cowlitz, Kalama, and Sandy rivers); historical counts, however, put these runs at more than 20,000 fish. In general, all runs in the ESU have declined over the past 20 years, with sharp declines in the last 5 years.

4.1.10.3 Habitat and Hydrology

Steelhead in this ESU are thought to use estuarine habitats extensively during outmigration, smoltification, and spawning migrations. The lower reaches of the Columbia River are highly modified by urbanization and dredging for navigation. The upland areas covered by this ESU are extensively logged, affecting water quality in the smaller streams used primarily by summer runs. In addition, all major tributaries used by LCR steelhead have some form of hydraulic barrier that impedes fish passage. Barriers range from impassible structures in the Sandy River basin that block access to extensive, historically occupied, steelhead habitat, to passable but disruptive projects on the Cowlitz and Lewis rivers. The Biological Review Team (1997) viewed the overall effect of hydrosystem activities on this ESU as an important determinant of extinction risk.

4.1.10.4 Hatchery Influence

Many populations of steelhead in the Lower Columbia River ESU are dominated by hatchery escapement. Roughly 500,000 hatchery-raised steelhead are released into drainages within this ESU each year. As a result, first-generation hatchery fish are thought to make up 50% to 80% of the fish counted on natural spawning grounds. The effect of hatchery fish is not uniform, however. Several runs are mostly hatchery strays (e.g., the winter run in the Cowlitz River [92%] and the Kalama River [77%], the summer run in the North Fork Washougal River [50%]), whereas others are almost free of hatchery influence (the summer run in the mainstem Washougal River [0%] and the winter runs in the North Fork Toutle and Wind rivers [0% to 1%]).

4.1.10.5 Other

Escapement estimates for the steelhead fishery in the Lower Columbia River ESU are based on inriver and estuary sport-fishing reports; there is a limited ocean fishery on this ESU. Harvest rates range from 20% to 50% on the total run, but for hatchery-wild differentiated stocks, harvest rates on wild fish have dropped to 0% to 4% in recent years (punch card data from WDFW through 1994).

4.1.10.6 Population Trends and Risks

For the LCR steelhead ESU as a whole, NMFS estimates that the average population growth rate (lambda) over the base period ranges from 0.98 to 0.36, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin (Table A-5a through A-5d; Appendix B in McClure et al. 2000). NMFS has also estimated the risk of absolute extinction for seven of the subbasin populations, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years ranges from zero for the Kalama River summer run and the Clackamas and Kalama River winter runs to 1.00 for the Clackamas River summer run and the Toutle River winter run (Table A-6a). Assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years rises to 1.00 for all but one population (the risk of extinction is 0.94 for the Green River winter run; Table A-6d).

NMFS has also calculated the proportional increase in the average growth rate of each of the seven subbasin populations that would be needed to reduce the risk of absolute extinction within 100 years to 5% (Tables A-6a through A-6d; Appendix B in McClure et al. 2000). Assuming that the effectiveness of hatchery fish has been zero, changes in population growth rates of less than 20% would be needed for each of the wild populations (Table A-6a). The needed change in growth rate rises to a maximum of 304%, for the Kalama River summer run, if hatchery-origin spawners have been 100% as effective as wild fish (Table A-6d).

4.1.11 Columbia River Chum Salmon

4.1.11.1 Geographic Boundaries and Spatial Distribution

Chum salmon of the Columbia River ESU spawn in tributaries and in a single known mainstem spawning area below Bonneville Dam. Most fish spawn on the Washington side of the Columbia River (Johnson et al. 1997).

4.1.11.2 Historical Information

Previously, chum salmon were reported in almost every river in the lower Columbia River basin, but most runs disappeared by the 1950s (Rich 1942, Marr 1943, Fulton 1970). Currently, the Washington Department of Fish and Wildlife (WDFW) regularly monitors only a few natural populations in the basin, one in Grays River, two in small streams near Bonneville Dam, and the mainstem area next to one of the latter two streams.

4.1.11.3 Life History

Chum salmon enter the Columbia River from mid-October through early December and spawn from early November to late December. Recent genetic analysis of fish from Hardy and Hamilton creeks and from the Grays River indicate that these fish are genetically distinct from other chum salmon populations in Washington. Genetic variability within and between populations in several geographic areas is similar, and populations in Washington show levels of genetic subdivision typical of those seen between summer- and fall-run populations in other areas and typical of populations within run types (Salo 1991, WDF et al. 1993, Phelps et al. 1994, and Johnson et al. 1997).

4.1.11.4 Other

Historically, the Columbia River chum salmon ESU supported a large commercial fishery, landing more than 500,000 fish per year. Commercial catches declined beginning in the mid-1950s. There are now no recreational or directed commercial fisheries for chum salmon in the Columbia River, although chum salmon are taken incidentally in the gill-net fisheries for coho and chinook salmon, and some tributaries have a minor recreational harvest (WDF et al. 1993).

4.1.11.5 Population Trends and Risks

Hatchery fish have had little influence on the wild component of the CR chum salmon ESU. NMFS estimates an average population growth rate (lambda) over the base period, for the ESU as a whole, of 1.03 (Table A-5a through A-5d; Appendix B in McClure et al. 2000). Because census data are peak counts (and because the precision of these counts decreases markedly during the spawning season as water levels and turbidity rise), the NMFS is unable to estimate the risk of absolute extinction for this ESU.

4.1.12 Snake River Sockeye Salmon

4.1.12.1 Geographic Boundaries and Spatial Distribution

The only remaining sockeye in the Snake River system are found in Redfish Lake, on the Salmon River. The nonanadromous form (kokanee), found in Redfish Lake and elsewhere in the Snake River basin, is included in the ESU.

4.1.12.2 Historical Information

Snake River sockeye were historically abundant in several lake systems of Idaho and Oregon. However, all populations have been extirpated in the past century except fish returning to Redfish Lake.

4.1.12.3 Life History

In general, juvenile sockeye salmon rear in the lake environment for 1, 2, or 3 years before migrating to sea. Adults typically return to the natal lake system to spawn after spending 1, 2, 3, or 4 years in the ocean (Gustafson et al. 1997).

4.1.12.4 Habitat and Hydrology

In 1910, impassable Sunbeam Dam was constructed 20 miles downstream of Redfish Lake. Although several fish ladders and a diversion tunnel were installed during subsequent decades, it is unclear whether enough fish passed above the dam to sustain the run. The dam was partly removed in 1934, after which Redfish Lake runs partially rebounded. Evidence is mixed as to whether the restored runs constitute anadromous forms that managed to persist during the dam years, nonanadromous forms that became migratory, or fish that strayed in from outside the ESU.

4.1.12.5 Population Trends and Risks

NMFS proposed an interim recovery level of 2,000 adult Snake River sockeye salmon in Redfish Lake and two other lakes in the Snake River basin (Table 1.3-1 in NMFS 1995_ Proposed Recovery Plan). The ESU contains less than 10 wild adults. Numbers are inadequate for a CRI-type analysis, but clearly the risk of extinction is very high.

4.2 SPECIES-LEVEL BIOLOGICAL REQUIREMENTS

The species-level biological requirements of the 12 listed ESUs are described in Section 1.3. NMFS has adopted the species-level biological requirements as its jeopardy standard.

4.3 SPECIES STATUS WITH RESPECT TO SPECIES-LEVEL BIOLOGICAL REQUIREMENTS

The current status of each species, as described in Section 4.1, indicates that the species-level biological requirements described in Section 1.3 are not being met for any of the 12 ESUs considered in this consultation. Improvements in survival rates (assessed over the entire life cycle) are necessary to meet species-level biological requirements in the future.